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# Impact of earthworm *Lumbricus terrestris* living sites on the greenhouse gas balance of no-till arable soil

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**Abstract.** We studied the effect of the deep-burrowing earthworm *Lumbricus terrestris* on the greenhouse gas (GHG) fluxes and global warming potential (GWP) of arable no-till soil using both field measurements and a controlled 15-week laboratory experiment. In the field, the emissions of nitrous oxide (N<sub>2</sub>O) and carbon dioxide (CO<sub>2</sub>) were on average 43 and 32 % higher in areas occupied by *L. terrestris* (the presence judged by the surface midden) than in adjacent, unoccupied areas (with no midden). The fluxes of methane (CH<sub>4</sub>) were variable and had no consistent difference between the midden and non-midden areas. Removing the midden did not affect soil N<sub>2</sub>O and CO<sub>2</sub> emissions. The laboratory results were consistent with the field observations in that the emissions of N<sub>2</sub>O and CO<sub>2</sub> were on average 27 and 13 % higher in mesocosms with than without *L. terrestris*. Higher emissions of N<sub>2</sub>O were most likely due to the higher content of mineral nitrogen and soil moisture under the middens, whereas *L. terrestris* respiration fully explained the observed increase in CO<sub>2</sub> emissions in the laboratory. In the field, the significantly elevated macrofaunal densities in the vicinity of middens likely contributed to the higher emissions from areas occupied by *L. terrestris*. The activity of *L. terrestris* increased the GWP of field and laboratory soil by 50 and 18 %, but only 6 and 2 % of this increase was due to the enhanced N<sub>2</sub>O emission. Our results suggest that high N<sub>2</sub>O emissions commonly observed in no-till soils can partly be explained by the abundance of *L. terrestris* under no-till management and that *L. terrestris* can markedly regulate the climatic effects of different cultivation practises.

## 1 Introduction

Agricultural soils can significantly contribute to the global greenhouse gas (GHG) exchange, but the contribution varies among the gases. For nitrous oxide (N<sub>2</sub>O), the emissions from agricultural soils account for 60 % of the anthropogenic emissions (Smith et al., 2007), whereas for methane (CH<sub>4</sub>), mineral agricultural soils are usually sinks as the aerobic topsoil favours methanotrophic bacteria (Hütsch, 2001). For carbon dioxide (CO<sub>2</sub>), soils can be either sinks or sources, depending on the balance of carbon input and output (Stockmann et al., 2013). N<sub>2</sub>O emissions are mainly regulated by soil oxygen status, but also by the availability of nitrogen and organic carbon (Granli and Bockman, 1994). The oxygen availability varies with soil structure and moisture and the potential for N<sub>2</sub>O emissions is greatest when the water-filled pore space (WFPS) is 60–70 % (Davidson, 1991) as this enables both nitrification and denitrification. When the WFPS is above 70 %, only denitrification takes place due to the shortage of oxygen and the dominating end product is the N<sub>2</sub> gas.

The application of no-till practice has recently increased in the agriculture (Derpsch et al., 2010). No-till often increases carbon sequestration to soils and is therefore considered as a useful cultivation technique in climate change mitigation (Lal, 1997). Elevated N<sub>2</sub>O emissions may, however, decrease the atmospheric benefits of no-till (Li et al., 2005; Sheehy et al., 2013; Palm et al., 2014) as the denser physical structure (Tebrügge and Düring, 1999; Schjønning and Rasmussen, 2000) and higher moisture content (e.g. Sharratt, 1996; Gregorich et al., 2008) of no-tilled soils lead to higher N<sub>2</sub>O emissions. The abundance and diversity of earthworms

can also be markedly higher under no-till than conventional tillage (Edwards and Lofty, 1982; Chan, 2001; Rothwell et al., 2011) and the role of earthworms in the regulation and enhancement of GHG emissions has recently gained increasing attention. Field results are still scarce, but a recent meta-analysis of laboratory studies suggests that the presence of earthworms can increase  $\text{N}_2\text{O}$  and  $\text{CO}_2$  emissions by 42 and 33 %, respectively (Lubbers et al., 2013a). A number of factors potentially contribute to this phenomenon. For instance, by burrowing, casting and mixing crop residues into the soil, the earthworms change soil organic carbon cycling, porosity, aggregation and gas diffusivity, enhance decomposition and increase the amount of mineral nitrogen in the soil (e.g. Subler and Kirsch, 1998; Lubbers et al., 2011). Earthworm casts and burrow linings also have higher microbial activity and more denitrifying bacteria than the bulk soil (Svensson et al., 1986; Brown et al., 2000; Elliott et al., 1990) and the moist anaerobic environment in the earthworm gut can stimulate microbial  $\text{N}_2\text{O}$  production (Karsten and Drake, 1997; Drake and Horn, 2006). On the other hand, earthworms can increase microaggregate formation and the stability of soil carbon (Fonte et al., 2007; Six and Paustian, 2014), and it is still unclear whether earthworms increase or decrease soil organic carbon stocks in the long term (Lubbers et al., 2013a; Blouin et al., 2013; Zhang et al., 2013).

Reduced tillage and no-till increase the densities of anecic, deep-burrowing earthworms in arable fields (Whalen and Fox, 2007). In the temperate and boreal fields, this group is mainly represented by the dew-worm, *Lumbricus terrestris* L. (Chan, 2001; Kladvik, 2001). In Finland, *L. terrestris* is the second most common earthworm species in arable fields, lagging only behind *Aporrectodea caliginosa* Sav. (Nieminen et al., 2011), and has the typical positive association with non-inversion cultivation (Nuutinen, 1992; Nuutinen et al., 2011). It is a large earthworm, which efficiently forages on crop residues (Subler and Kirsch, 1998; Shuster et al., 2000) and builds middens (i.e. small mounds of collected litter and surface castings) at the openings of its permanent burrows, often penetrating deeper than 1 m (e.g. Nuutinen and Butt, 2003). The middens are biological hot spots with high microbial activity (Schrader and Seibel, 2001; Aira et al., 2009), diverse invertebrate populations (Hamilton and Sillman, 1989; Maraun et al., 1999; Butt and Lowe, 2007) and higher nutrient and organic carbon contents than the surrounding soil (Subler and Kirsch, 1998; Wilcox et al., 2002; Aira et al., 2009). By transferring plant litter into the subsoil, *L. terrestris* may also increase the subsoil carbon stocks; for example, Don et al. (2008) estimated that *L. terrestris* sequesters carbon in the burrow linings at the rate of  $22 \text{ g C m}^{-2} \text{ yr}^{-1}$ . On the other hand, the turnover time of burrow wall carbon can be only 3–5 years (Don et al., 2008). This is because the well-aerated burrow walls allow the expansion of high microbial activity down the soil profile (Loquet et al., 1977 in Devliegher and Verstraete, 1997) and the interactions among microbes and their feeders in the burrow walls are intense

and accelerate carbon and nutrient mineralization (Tiunov and Scheu, 1999; Görres et al., 1999, 2001). The burrows of *L. terrestris* are also bypass flow routes for percolating water, and depending on arable soil management, they may increase leaching of topsoil nitrogen to the subsoil (Shuster et al., 2003).

Most of the investigations of earthworm effects on GHG emissions have been carried out in the laboratory (Bertora et al., 2007; Rizhiya et al., 2007; Giannopoulos et al., 2010; Lubbers et al., 2011; Augustenborg et al., 2012) and to our knowledge, only three field experiments have been conducted (Borken et al., 2000; Amador and Avizis, 2013; Lubbers et al., 2013b). Recent reviews have underlined the need for field studies with all major gases ( $\text{N}_2\text{O}$ ,  $\text{CO}_2$  and  $\text{CH}_4$ ) to provide a more comprehensive picture of earthworm contribution to soil GHG emissions (Lubbers et al., 2013a; Blouin et al., 2013). In this study, we aimed at filling this research gap by measuring the small-scale spatial variation of soil biological and chemical properties and  $\text{N}_2\text{O}$ ,  $\text{CO}_2$  and  $\text{CH}_4$  fluxes caused by *L. terrestris* in a northern, arable no-till field. We hypothesized that (1) the  $\text{N}_2\text{O}$  and  $\text{CO}_2$  emissions are greater in *L. terrestris* midden areas (higher earthworm activity) compared to adjacent non-midden areas (lower earthworm activity), while  $\text{CH}_4$  emissions remain unaffected; (2) the middens contribute to gas production and their removal from the soil surface decreases instant gas emissions; and (3) the biological and chemical soil properties essential for gas balance differ between the midden and non-midden areas. Moreover, to test how well the earthworm effects on GHG emissions in the field can be predicted by laboratory experiments, we established a controlled laboratory study with a *L. terrestris* treatment and measurements of response variables identical to those in the field. Our aim was not to establish a laboratory experiment that would perfectly mimic our field situation, but to establish a typical laboratory experiment to test whether laboratory studies in general can produce results that resemble the field results. This is an important aspect as most earlier experiments have been carried out in the laboratory and e.g. the review by Lubbers et al. (2013a) is entirely based on laboratory studies.

## 2 Methods

### 2.1 Field measurements

Field measurements of  $\text{N}_2\text{O}$ ,  $\text{CO}_2$  and  $\text{CH}_4$  emissions were conducted in a long-term, no-till field (11 years of no-till cultivation) in Säkylä (60°58'N, 22°31'E), south-western Finland, in October 2008. The soil at the site (depth 0–20 cm) is fine sand with 15 % clay, 29 % silt and 56 % sand. Soil pH ( $\text{H}_2\text{O}$ ) is 6.1 and the N and C concentrations 0.1 and 2.1 %, respectively. The topsoil (0–5 cm) bulk density is  $1.37 \text{ g cm}^{-3}$ . The annual crops cultivated in the field in 2007 and 2008 were turnip rape and barley, respectively. Ten

large middens and their adjacent non-midden areas were randomly chosen within two 20 m<sup>2</sup> areas (called sites A and B; five pairs in both) 1 month after crop harvest, which according to our experience is a time of high *L. terrestris* activity. The two sites, 30 m apart, were needed to obtain a sufficient number of treatment pairs, but they also provide data for testing whether the treatment effect varies in space at the field scale. For this purpose, the site was included in the statistical models as an explaining factor. In order to minimize the environmental variation within treatment pairs, the distance between the midden and non-midden areas within a pair was kept short; the average distance between the outer rims of measurement chambers within a pair was 13 cm (min 3 cm, max 34 cm), while the average distance between a pair and its closest counterpart was 1.35 m (min 0.37 m, max 3.00 m).

The gas measurements were accomplished using round PVC chambers (diameter 15 cm, height 10 cm). Five gas measurements were carried out at varying intervals over a period of 2 weeks. Chambers were pressed into the soil to the depth of approximately 2 cm and the soil was compressed by hand around the chambers. Permanent installations were not established in order to avoid the disturbance of earthworms, and since the experiment was conducted after harvest, it was not necessary to take into account the decrease of CO<sub>2</sub> flux that may follow when live roots are cut by the chamber (see Heinemeyer et al., 2011). In each measurement, 20 mL of chamber air was sampled through a rubber septum using a polypropylene syringe (BD Plastipak, Becton, Dickinson and Company, Franklin Lakes, NJ, USA) immediately and 60 min after the placement of the chamber. The air was then transferred into pre-evacuated 12 mL glass vials (Exetainer, Labco Ltd., High Wycombe, UK). Before each gas sample, the air in the chamber was mixed by one syringe flush.

The air temperature, which was measured using a Fluke 52 II thermometer (Fluke Corp., USA), fluctuated between 7.2 and 11.8 °C during the gas measurements. Air temperature, instead of the chamber temperature, was used to define the gas volume for flux calculation as chamber warming due to radiation is minimal in October. Soil moisture was measured next to each “midden–non-midden” pair at the depth of 0–15 cm during each gas measurement using a TRASE system I moisture meter and time domain reflectometry (TDR; Soil Moisture Equipment Corp., Goleta, CA, USA). The changes in soil temperature were followed using thermologgers (ElcoLog, Elcoplast Oy, Finland), which were installed at the depth of 5 cm outside the gas sampling areas (this data is missing for the two first gas measurements).

At the last measurement, gas samples were first taken as described above. The middens (surface cast mounds and the associated residues) and the straw litter of the non-midden areas were then removed and the gas measurements were repeated to evaluate the effect of midden and straw material on gas emissions. After these measurements, soil cores (diameter 5 cm, depth 5 cm) were collected from the entrance of *L. terrestris* burrows and the adjacent non-midden areas.

The removed midden and straw material and the soil samples were stored at –18 °C for 7.5 months before being analysed for gravimetric moisture content, potential denitrification and mineral N concentrations. To estimate earthworm abundances in the area of the gas measurement, the measurement chamber was pushed deeper into the soil and the earthworms were hand-sorted out of the obtained soil sample (diameter 15 cm, depth 15 cm). Deep-residing earthworms were extracted from the bottom of the pit by pouring 0.5–0.75 L formalin solution (0.5 %) into the pit and collecting individuals that emerged within 30 min. Slugs, which were abundant in the middens, were hand-sorted from the midden and non-midden area samples and together with the earthworms were stored in 85 % ethanol, weighted and identified into the species or genus level (Sims and Gerard, 1999; Kerney and Cameron, 1979).

## 2.2 Laboratory experiment

The soil, barley stubble straw and *L. terrestris* individuals were collected for the laboratory mesocosms in the beginning of November 2008 from the same no-till field that was used for field measurements. The 15-week experiment was designed to simulate the post-harvest autumn conditions of a no-till field and during the set-up; all unnecessary manipulation of soil, straw and earthworms was avoided to preserve the natural communities of microbes and soil micro- and meso-fauna. The moist soil (moisture content 27 % of fresh mass) was first sieved (6 mm) and mixed to ensure soil homogeneity. Any earthworms found were removed. Thirty PVC tubes (diameter 15 cm, height 45 cm, bottoms enclosed with plastic lids) were then filled with the soil to the height of 43 cm. During filling, the soil was compacted to achieve even bulk density among the tubes (mean 1.43 g cm<sup>–3</sup>, min 1.40 and max 1.46 g cm<sup>–3</sup>,  $n = 30$ ). The tubes were weighted (before and after filling) and placed in an incubation room at 15–17 °C, chosen as favourable temperature for *L. terrestris* activity (Butt, 1991), with a rhythm of 10 h day (fluorescent lamps providing on average 1102 lx) and 14 h night (no illumination). Air humidity was maintained using a moistener, but varied from 26 to 81 % during the experiment. Soil moisture content was adjusted to 28 % and kept approximately constant by adding deionized water once a week (always 2 days before gas samplings) and spraying the soil surface with water after gas measurements.

The *L. terrestris* individuals used in the experiment were extracted from the field using a mustard mixture (Gunn, 1992) and immediately washed in tap water. Individuals were kept in moist soil for 9 days (dark, 4 °C) before one large individual was added to each of the 15 randomly chosen mesocosms. Each individual was weighted (mean fresh mass 4.5 g, min 3.7 g, max 5.5 g) and the settling into the soil was facilitated by creating an artificial burrow (depth 8.5 cm, diameter 0.5 cm) in the centre of the soil column. The remaining 15 mesocosms were left without worms and served as

controls. The *L. terrestris* and control mesocosms were randomly placed in the incubation room as treatment pairs. An even layer of chopped straw was added on the top of the soil in each mesocosm (straw length 2 cm, total fresh mass 5 g), and to prevent animal escape, the mesocosms were covered by a mesh. Emerging plant seedlings were removed from the mesocosms during the experiment, whereas juvenile earthworms, noticed to hatch from the cocoons, were not, as the removal would have disturbed the experiment.

The gas measurements were started 1 month after mesocosm establishment and were repeated twelve times, at 1-week intervals, from December 2008 to February 2009. The sampling was always carried out within 1 day. For the measurements, air-proof plastic lids (diameter 15 cm, height 10 cm) were first placed on the tubes air-tightly. The incubation lasted for 60 min and the samples were collected according to the field protocol described above. At the final date, gas fluxes were measured before and after removing *L. terrestris* midden and straw residues. The soil samples for soil moisture, potential denitrification and mineral N measurements were taken as in the field. The tubes were emptied and the *L. terrestris* individuals and earthworm juveniles, hatched from the cocoons during the experiment, were hand-sorted out of the soil. A 100 g subsample was taken from the mixed soil to estimate the mineral N content of the entire soil column. At the end of the experiment, three of the *L. terrestris* mesocosms had 1–3 and seven of the control mesocosms 1–2 small earthworm juveniles (both dark and light pigmented unidentified species) having a maximum individual fresh mass of 0.16 g. All earthworms were washed in deionized water and weighted and, in order to determine their GHG production, incubated in 210 mL flasks for 60 min (separately for experimental *L. terrestris* and the group of juveniles). The GHG production was estimated using 10 mL gas samples taken in the beginning and at the end of the incubation. Three incubations of *L. terrestris* produced deviant fluxes of  $\text{N}_2\text{O}$ ,  $\text{CO}_2$  and  $\text{CH}_4$ , and the results were excluded from the data set.

### 2.3 Analyses of gases, potential denitrification and mineral nitrogen

The gas samples were always analysed within 48 h after sampling using a gas chromatograph (GC) equipped with a flame ionizer (FID), an electron capture detector (ECD) and a nickel catalyst for converting  $\text{CO}_2$  to  $\text{CH}_4$ . The precolumn and analytical columns consisted of 1.8 and 3 m long steel columns, respectively, packed with 80/100 mesh Haysep Q (Supelco Inc., Bellefonte, PA, USA). The GC (HP 6890 Series, GC System, Hewlett Packard, USA) had a 10-way valve with a 2 mL sample loop and a backflush system for separating water from the sample and for flushing the precolumn between the runs. A six-way valve was used to lead the flow to either the FID or ECD. The temperature of the GC oven, FID and ECD was 70, 300 and 350 °C, respectively. Nitro-

gen was used as the carrier gas and a mixture of argon and methane (5 %) as a make-up gas ( $1.4 \text{ mL min}^{-1}$ ) to increase the ECD sensitivity. A standard gas mixture (AGA Gas AB, Lidingö, Sweden) of known  $\text{N}_2\text{O}$ ,  $\text{CH}_4$  and  $\text{CO}_2$  concentrations was used for the calibration curve. The flux rate of each gas was calculated using the gas accumulation rate during the 60 min enclosure period. Cumulative fluxes were calculated by assuming linear changes between subsequent measurement dates. The net gas balance as a global warming potential (GWP) was determined using the factor 298 for  $\text{N}_2\text{O}$  and 25 for  $\text{CH}_4$  (Myhre et al., 2013).

The denitrification potentials of the midden soil and the straw of the *L. terrestris* middens and the adjacent non-midden areas were determined as in Klemetsson et al. (1988) and Henault et al. (1998) with some modifications. In brief, the defrosted and sieved 10 g (d.m.) soil samples (moisture was on average 26 % in the field and 21 % in the laboratory samples) were placed in 120 mL bottles and 4 mL of distilled water was added. The straw samples were combined within treatments (midden vs. non-midden, separately for areas A and B), because the amount of material in one sample was not enough for the analysis, and then divided into 2.5–5.5 g (d.m.) subsamples. After one night at 6 °C, the samples were transferred to 25 °C and treated with 5 mL of potassium nitrate ( $\text{KNO}_3$ ) solution and 5 mL of glucose solution (corresponding to amendments of 200 mg N and 500 mg C  $\text{kg}^{-1}$  soil). The bottles were then sealed using butyl rubber septa and crimp seals, evacuated, and flushed three times with dinitrogen gas. The overpressure in the bottles was released through a 0.5 mm needle, pierced through the septum, and to prevent the entry of oxygen into the bottle, the needle was mounted on a 1 mL plastic syringe (without piston) filled with 0.1 mL distilled water. The bottles were then amended with 12 mL of acetylene ( $\text{C}_2\text{H}_2$ ) to block the  $\text{N}_2\text{O}$  reduction step of denitrification, which was regarded as the start of the incubation ( $t = 0$ ). Three mL gas samples were then taken after 15 and 45 min, followed by 1 mL samples after 75, 105, 135, 165, 195, 225 and 255 min, and these were injected into 12 mL evacuated vials. All samples were diluted with  $\text{N}_2$  to a volume of 18 mL to ensure that the concentrations were in the range of the calibration curve. Samples were analysed using the Hewlett Packard GC as described above.

For the analyses of soil ammonium and nitrate concentrations, samples were first homogenized manually using a steel spatula, and from each sample 50 g of fresh soil was mixed with 125 mL of 2 M KCl and shaken for 2 h on an orbital shaker. The amount of straw material in one sample was too small for the analysis, so straw samples were combined within treatments. The combined samples were then divided into 6–21 g (f.w.) subsamples and treated similarly as the soil samples. The extracts of soil and straw samples were filtered through filter paper ( $130 \text{ g m}^{-2}$ , Tervakoski Oy, Tervakoski, Finland) and analysed for nitrate and ammonium the next day after storage at 6 °C. A colorimetric autoanalyser (QuikChem

AE, Lachat Instruments, Loveland, CO, USA) was used for the simultaneous analysis of nitrate and ammonium.

## 2.4 Statistical analyses

The field data of N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> emissions were obtained from a randomized complete block design with repeated measurements. Altogether, there were ten pairs (blocking factor) of midden–non-midden areas (treatment factor) from the two sites (A and B). The measurements at the same experimental site were correlated, which was taken into account in the statistical models through appropriate covariance structures. The statistical model thus became

$$y_{ijkl} = \mu + s_i + \beta_{j(i)} + t_k + (st)_{ik} + \varepsilon_{ijk} + d_l + (sd)_{il} + (\beta d)_{jl(i)} + (td)_{kl} + (std)_{ikl} + \gamma_{ijkl}, \quad (1)$$

where  $\mu$  is the constant intercept, and  $s_i$ ,  $t_k$ ,  $(st)_{ik}$ ,  $d_l$ ,  $(sd)_{il}$ ,  $(td)_{kl}$  and  $(std)_{ikl}$  are fixed main and interaction effects for site ( $s$ ), treatment ( $t$ ) and date ( $d$ ). The  $\beta_{j(i)}$  is the random effect for block  $j$  within site  $i$  and  $\varepsilon_{ijk}$  is random plot-to-plot variation, all mutually independent with variances  $\text{var}(\beta_{j(i)}) = \sigma^2_\beta$  and  $\text{var}(\varepsilon_{ijk}) = \sigma^2_\varepsilon$ . The  $(\beta d)_{jl(i)}$  represents the random date-specific contribution for block  $i$  within site  $j$ , and  $\gamma_{ijkl}$  represents the random error effect for observations on the same plot (Gumpertz and Brownie, 1993). This model was used for CH<sub>4</sub>. For N<sub>2</sub>O and CO<sub>2</sub>, a simplified model was used as the site had no effect on the fluxes of either gas. The effect of removing middens and straw litter from the soil surface on N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> emissions was analysed using a similar model as for the repeated gas measurements, except that the repeated measurement effect of date was replaced with the repeated measurement effect of before and after removal. Analogously to the earlier models, the site effect was included in the model for CH<sub>4</sub>, but not for N<sub>2</sub>O and CO<sub>2</sub>. In the case of N<sub>2</sub>O, log transformation was used to meet the normality assumption.

The background variables were measured at the last measurement date (Table 4). Since these measurements were not repeated, the statistical models used were simplified analogues of the model presented above, except for the number of slugs, which was analysed using the non-parametric Wilcoxon sign rank test as the assumptions of the parametric methods were not met. The cumulative emissions of N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> were analysed using a simplified non-repeated analogue of the model presented above. The analysis of laboratory data followed the analysis of field data, except that the site effect and interactions were not included in the models. Log transformations were used for N<sub>2</sub>O and mineral nitrogen (top 5 cm soil samples) and in addition, two outliers were excluded from the mineral nitrogen data due to exceptionally high values in comparison to the other 13 observations in the control mesocosms.

For all the parametric models, REML (restricted maximum likelihood) was used as the estimation method, degrees of freedom were calculated by the Kenward–Roger method

(Kenward and Roger, 1997), and model assumptions were checked using appropriate graphs. The models were fitted using the MIXED procedure of SAS 9.2 (SAS Institute Inc., Cary, NC, USA) and pairwise comparisons were performed using two-sided  $t$ -type tests.

## 3 Results

### 3.1 Field measurements

In the field, the N<sub>2</sub>O and CO<sub>2</sub> emissions were significantly higher in the midden than non-midden areas (Table 1; Fig. 1a, b). The overall (all repeated measurements included) model-based mean estimates of N<sub>2</sub>O fluxes were 0.23 (95 % CI 0.18–0.27) and 0.13 (0.09–0.17)  $\mu\text{g N chamber area}^{-1} \text{ h}^{-1}$  for the midden and non-midden areas, respectively. The corresponding figures for CO<sub>2</sub> were 1754 (1568–1941) and 1201 (1015–1388)  $\mu\text{g CO}_2 \text{ chamber area}^{-1} \text{ h}^{-1}$ , respectively. Based on these estimates, the chamber area with one midden produced on average 43 % more N<sub>2</sub>O and 32 % more CO<sub>2</sub> than an equivalent non-midden area. N<sub>2</sub>O and CO<sub>2</sub> emissions varied among the dates (Fig. 1a, b; Table 1), but this variation was apparently not explained by soil moisture or temperature, which fluctuated little among the dates (min–max 37.2–38.3 % and 6.5–8.5 °C, respectively). The CH<sub>4</sub> fluxes differed between the midden and non-midden areas at two measurement dates, but the effects were specific to the measurement site (Table 1), i.e. the flux was higher in the midden than non-midden areas in site B at the first measurement ( $t_{14,1} = -4.02$ ,  $p = 0.001$ ), but lower in site A at the fourth measurement ( $t_{12,4} = 2.44$ ,  $p = 0.031$ ; Fig. 1c, d). The model-based mean estimates of cumulative emissions were significantly higher in the midden than non-midden areas for N<sub>2</sub>O and CO<sub>2</sub> ( $F_{1,7.34} = 16.91$ ,  $p = 0.004$ ;  $F_{1,7.66} = 43.80$ ,  $p < 0.001$ , respectively), but not for CH<sub>4</sub> ( $F_{1,7.74} = 3.24$ ,  $p = 0.111$ ) (Table 2). The removal of middens and other residues from the soil surface had no effect on N<sub>2</sub>O and CO<sub>2</sub> emissions in either the midden or non-midden areas (Table 3; Fig. 1a, b). For CH<sub>4</sub>, the removal decreased the flux in site A ( $t_{9,1} = 2.86$ ,  $p = 0.019$ ), but not in site B ( $t_{7,87} = -0.65$ ,  $p = 0.532$ ), and no difference was found between the responses of midden and non-midden areas (Table 3, Fig. 1c, d).

The number of earthworms was 125 % and their biomass 150 % higher in the midden than in the non-midden areas (Table 4). However, only in four midden and two non-midden areas, a large ( $> 0.8 \text{ g}$ ) *L. terrestris* was found and the majority of earthworms were juveniles. In the midden areas, 18 % of individuals belonged to *Lumbricus*, 51 % to *Aporrectodea* and 31 % remained unidentified. In the non-midden areas, the corresponding figures were 16, 58 and 26 %, respectively. The soil surrounding the burrow entrance (within 5 cm diameter) was on average 1 % unit moister, contained 23 % more mineral N and had 20 % higher potential denitrification

**Table 1.** Fixed effect (treatment and site) *P* values of general linear mixed models with repeated measurements (date) for N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> emissions in the field and laboratory measurements. Treatment is “midden area vs. non-midden area” in the field and “*L. terrestris* vs. control” in the laboratory mesocosms.

	Model term	N <sub>2</sub> O	CO <sub>2</sub>	CH <sub>4</sub>
Field	Site			0.008
	Treatment	<0.001	<0.001	0.043
	Treatment × site			0.072
	Date	0.004	<0.001	0.029
	Site × date		<0.001	
	Treatment × date	0.289	0.588	<0.001
	Treatment × site × date			0.007
Laboratory	Treatment	<0.0001	<0.0001	0.482
	Date	<0.0001	<0.0001	0.144
	Treatment × date	0.159	0.401	0.039

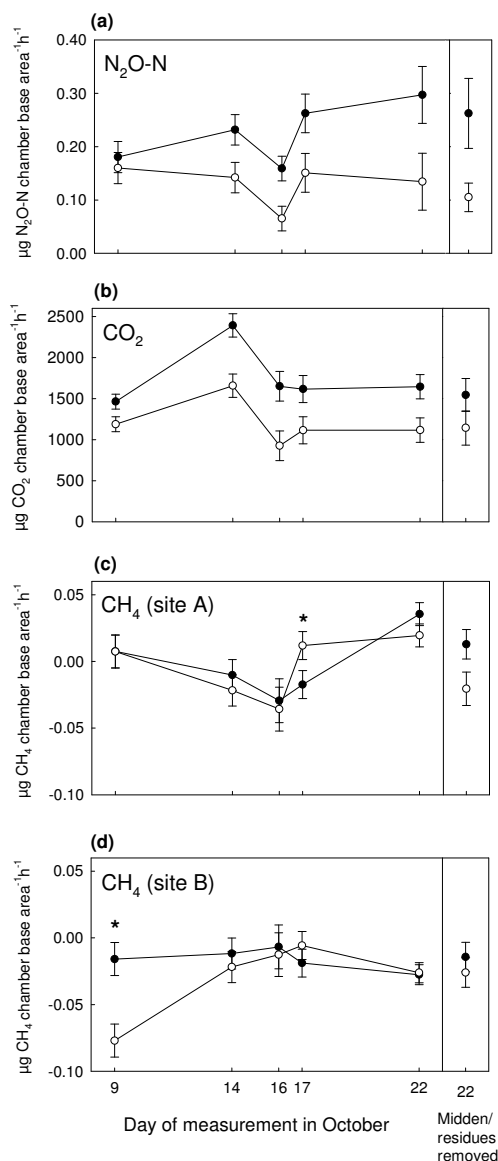
**Table 2.** The mean estimates (SE) of statistical models for cumulative N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> fluxes in the field (duration 2 weeks) and laboratory (15 weeks) measurements.

	N <sub>2</sub> O μg N chamber area <sup>-1</sup>	CO <sub>2</sub> mg chamber area <sup>-1</sup>	CH <sub>4</sub> μg chamber area <sup>-1</sup>
Field:			
Midden area	74.2 (5.1)	591.4 (28.4)	−2.6 (1.1)
Non-midden area	47.6 (5.1)	394.4 (28.4)	−4.8 (1.1)
Laboratory:			
<i>L. terrestris</i>	111.3 (7.1)	3224 (157)	−230.7 (9.2)
Control	90.3 (6.2)	2729 (152)	−224.7 (8.1)

than the topsoil of the non-midden areas (Table 4), but the denitrification potential of the midden and non-midden straw did not differ (2.7 vs. 2.8 μg N<sub>2</sub>O–N g<sup>−1</sup> straw d.m. h<sup>−1</sup>, respectively). The mineral N content of the straw was 28 and 69 mg kg<sup>−1</sup> straw d.m. in the midden and non-midden areas, respectively, while the midden areas had more straw litter on the soil surface (visual observation). In total, 31 slugs (*Arion fasciatus* N.) were found from the midden areas after the final gas measurement, while only three were found from the non-midden areas (Table 4). In the midden areas, 77 % of the slugs were found in the midden, 23 % in the soil beneath the midden.

### 3.2 Laboratory experiment

In the laboratory, N<sub>2</sub>O and CO<sub>2</sub> emissions were significantly higher with than without *L. terrestris* (Table 1; Fig. 2a, b). The model-based mean estimates (with all repeated measurements included) of N<sub>2</sub>O emissions with and without *L. terrestris* were 0.060 (95 % CI 0.053–0.067) and 0.044 (0.039–0.049) μg N chamber base area<sup>−1</sup> h<sup>−1</sup>. The corresponding figures for CO<sub>2</sub> flux were 1769 (1600–1937) and 1536 (1367–1704) μg CO<sub>2</sub> chamber base area<sup>−1</sup> h<sup>−1</sup>, respectively. Based on these values, one *L. terrestris* individual increased the mesocosm emission of N<sub>2</sub>O and CO<sub>2</sub> by 27 and 13 %, respectively. On average, the fluxes of N<sub>2</sub>O and CO<sub>2</sub> decreased in the course of the experiment (Fig. 2a, b). The CH<sub>4</sub> flux fluctuated during the experiment without a clear



**Figure 1.** The mean ( $\pm$ SE) estimates of statistical models for (a) N<sub>2</sub>O, (b) CO<sub>2</sub> and (c, d) CH<sub>4</sub> (separately for field sites A and B) emissions in *L. terrestris* midden (●) and non-midden (○) areas and the effect of the removal of middens and surface residues on the emissions. For CH<sub>4</sub>, the differences between the midden and non-midden areas at  $p < 0.05$  are marked with \* (for effects on N<sub>2</sub>O and CO<sub>2</sub>, see Table 1).

trend (Table 1b, Fig. 2c) and, only at day 98, the emission rate differed between the treatments, being then higher with than without *L. terrestris* ( $t_{171} = -2.12$ ,  $p = 0.035$ ). The model-based mean estimates of the cumulative emissions were significantly higher with than without *L. terrestris* for N<sub>2</sub>O and CO<sub>2</sub> ( $F_{1,12.9} = 5.09$ ,  $p = 0.042$ ;  $F_{1,9.65} = 29.21$ ,  $p < 0.001$ , respectively), but not for CH<sub>4</sub> ( $F_{1,11.5} = 0.33$ ,  $p = 0.579$ ) (Table 2).

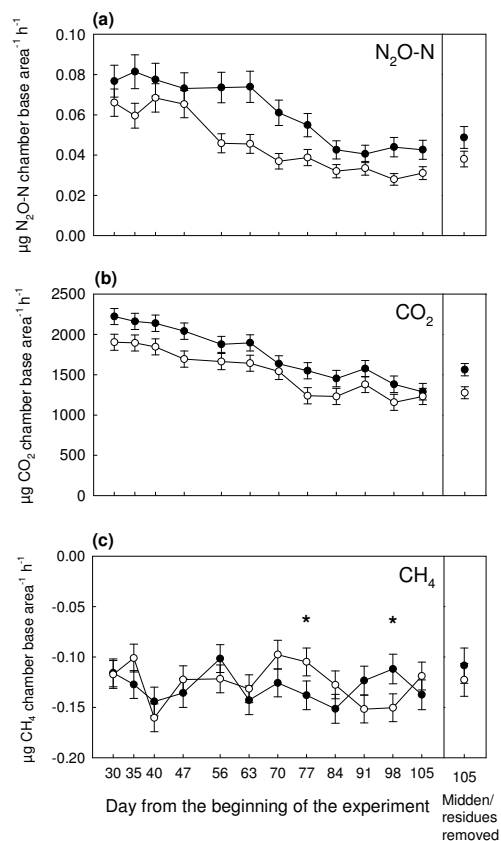
**Table 3.** Fixed effect (site and treatment) *P* values of general linear mixed models with repeated measurements (midden and residue removal) for N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> emissions in the field and laboratory measurements. Treatment is “midden area vs. non-midden area” in the field and “*L. terrestris* vs. control” in the laboratory mesocosms.

	Model term	N <sub>2</sub> O	CO <sub>2</sub>	CH <sub>4</sub>
Field	Site			0.007
	Treatment	0.012	0.009	0.015
	Treatment × site			0.080
	Removal	0.401	0.980	0.139
	Site × removal			0.034
	Treatment × removal	0.845	0.338	0.176
	Treatment × site × removal			0.894
	Removal	0.083	0.002	0.886
Laboratory	Treatment	0.004	0.008	0.440
	Removal	0.449	0.054	0.317
	Treatment × removal			

The removal of middens and straw residues from the soil surface affected the N<sub>2</sub>O and CO<sub>2</sub> emissions, but not the CH<sub>4</sub> emissions (Table 3; Fig. 2a–c). The N<sub>2</sub>O emissions increased after the removal in all mesocosms, whereas the response of CO<sub>2</sub> flux depended on the treatment: the removal increased CO<sub>2</sub> emissions in the presence ( $t_{26} = -3.36$ ,  $p = 0.002$ ), but had no effect in the absence of *L. terrestris* ( $t_{26} = -0.64$ ,  $p = 0.525$ ).

At the end of the experiment, mesocosms with *L. terrestris* had less straw litter on the soil surface (visual observation) and 4 % more mineral N in the 0–43 cm soil column (excluding the soil core collected around the burrow) than the mesocosms without *L. terrestris* (Table 5). In all except two mesocosms the resident worm had created a burrow that reached the bottom of the soil column. The soil that surrounded the *L. terrestris* burrow entrance (diameter 5 cm) was 0.3 % unit moister, contained 16 % more mineral N and had a 17 % greater potential denitrification rate than the topsoil of the control treatment (Table 5). The potential denitrification of the straw collected from *L. terrestris* and control mesocosms was 0.24 and 0.19  $\mu\text{g N}_2\text{O-N g}^{-1}$  straw d.m. h<sup>-1</sup> and its mineral N content 664 and 122 mg kg<sup>-1</sup> d.m., respectively.

Two of the 15 *L. terrestris* individuals had died and the remaining 13 had lost on average 1.0 g or 22 % weight during the 15-week experiment. When incubated in glass flasks at the end of the experiment, the mean emission rate of one *L. terrestris* individual (mean fresh mass 3.6 g, min 3.1 g and max 4.2 g) was 0.006 (SE 0.001)  $\mu\text{g N}_2\text{O-N}$ , 425 (41)  $\mu\text{g CO}_2$  and  $-0.001$  (0.002)  $\mu\text{g CH}_4 \text{ h}^{-1}$ . Mean emissions per unit fresh mass (min, max) for the three gases were 0.06 (0.03, 0.12), 2678 (1501, 4197) and  $-0.03$  ( $-0.19$ , 0.12) nmol gas g<sup>-1</sup> f.w. h<sup>-1</sup>, respectively. Based on these values, the proportion emitted by *L. terrestris* of the total N<sub>2</sub>O, CO<sub>2</sub> and CH<sub>4</sub> fluxes at the last gas measurement was 16, 36 and 0.7 %, respectively.



**Figure 2.** The mean estimates ( $\pm$ SE) of statistical models for (a) N<sub>2</sub>O, (b) CO<sub>2</sub> and (c) CH<sub>4</sub> emissions in *L. terrestris* (●) and control (○) mesocosms and the effect of the removal of middens and surface residues on the emissions. For CH<sub>4</sub>, the differences between treatments at  $p < 0.1$  are marked with \* (for effects on N<sub>2</sub>O and CO<sub>2</sub>, see Table 1).

## 4 Discussion

In agreement with our first hypothesis, field N<sub>2</sub>O and CO<sub>2</sub> emissions were greater in *L. terrestris* midden than non-midden areas. CH<sub>4</sub> fluxes were variable without a clear effect, but there was a slight indication that the presence of *L. terrestris* decreased the CH<sub>4</sub> oxidation rate of the soil. Against our second hypothesis, the removal of middens and residues from the soil surface did not decrease N<sub>2</sub>O and CO<sub>2</sub> emissions. This indicates that the effect of *L. terrestris* on GHG emissions results from changes in soil conditions at its living site, not from the surface midden. Following our third hypothesis, most of the investigated biological, chemical and physical soil variables differed between the midden and non-midden areas, telling of the significance of *L. terrestris* as an ecosystem engineer in arable fields. The fact that we found an equally positive effect of *L. terrestris* on N<sub>2</sub>O and CO<sub>2</sub> emissions in the laboratory further indicates that the observed effects in the field cannot be purely explained by confounding factors such as the burrows acting as a chimney for gas emis-

**Table 4.** Characteristics of *L. terrestris* midden ( $n = 10$ ) and adjacent non-midden ( $n = 10$ ) areas at the end of the field measurements (model-based mean estimates with 95 % confidence intervals presented for all other variables except for the slug *Arion fasciatus*, which has medians with a minimum and maximum). *F*- and *P*-statistics show the statistical significance of the difference between the midden and non-midden areas (for slugs the values are from the non-parametric Wilcoxon signed rank test).

	Midden area	Non-midden area	<i>df</i>	<i>F</i>	<i>P</i>
Earthworm number <sup>a</sup>	3.6 (2.6–4.6)	1.6 (0.6–2.6)	1, 8	8.51	0.019
Earthworm mass (g f.w.) <sup>a</sup>	2.0 (1.4–2.7)	0.8 (0.1–1.5)	1, 16	7.81	0.013
Slug number <sup>a</sup>	3.0 (0, 6)	0 (0, 1)		22.5	0.004
Soil moisture (% of f.w.) <sup>b</sup>	26.5 (25.8–27.2)	25.4 (24.8–26.1)	1, 8	7.66	0.024
Mineral N (mg kg <sup>-1</sup> soil d.w.) <sup>b</sup>	9.2 (7.9–10.5)	7.1 (5.7–8.4)	1, 8	8.24	0.021
Potential denitrification ( $\mu\text{g N}_2\text{O-N g}^{-1}$ soil d.w. h <sup>-1</sup> ) <sup>b</sup>	1.2 (1.1–1.4)	1.0 (0.9–1.2)	1, 8	4.16	0.076

<sup>a</sup> Sample covers the chamber base area (diameter 15 cm). <sup>b</sup> Soil core (depth 5 cm, diameter 5 cm) in the midden area taken around the *L. terrestris* burrow entrance.

sions from a larger area than the chamber, the worms selecting sites of high microbial activity, or *L. terrestris* affecting the emissions of the adjacent control area by collecting straw from it. However, the magnitude of the effect was significantly smaller in the laboratory than in the field, i.e. a 27 % vs. 43 % increase for N<sub>2</sub>O and 13 % vs. 32 % increase for CO<sub>2</sub>. It also appeared that the laboratory test could not fully simulate the role of *L. terrestris* middens in gas emissions as the removal of middens increased the emissions. These results underline the value of comparing the measurements in the laboratory to those in natural field sites with established earthworm populations.

Our results show that *L. terrestris* can create sites of elevated N<sub>2</sub>O emissions in arable no-till soils: in the field, the cumulative N<sub>2</sub>O emissions were 36 % higher in the midden than non-midden areas and, in the laboratory, 19 % higher in mesocosms with than without *L. terrestris*. These results are in good agreement with earlier laboratory studies (e.g. Matthies et al., 1999; Giannopoulos et al., 2010), but also with field studies, such as the study by Borken et al. (2000), which reported a 57 % increase in N<sub>2</sub>O emissions in beech forest mesocosms due to *L. terrestris*. The recent meta-analysis of laboratory studies by Lubbers et al. (2013a) also suggested a 42 % increase in soil N<sub>2</sub>O emissions in the presence of earthworms. Few opposite findings exist (e.g. Speratti and Whalen, 2008), although some studies suggest that the contribution of earthworms to N<sub>2</sub>O emissions could be transient (Amador and Avizinis, 2013; Lubbers et al., 2013b). In general, the contribution of earthworms to GHG emissions is composed of direct and indirect emissions. Direct emissions originate from earthworm metabolism and indirect from those changes the earthworms induce in their environment. Living earthworms have been found to emit N<sub>2</sub>O (Drake et al., 2006; Karsten and Drake, 1997) and our incubation measurements support these findings (Table 6). The reported values of direct N<sub>2</sub>O emissions emitted by *L. terrestris* vary from 0.05 to 0.95 nmol N<sub>2</sub>O-N g<sup>-1</sup> f.w. h<sup>-1</sup> (Matthies et al., 1999; Horn et al., 2006; Wüst et al., 2009), so

our value, 0.06 nmol of N<sub>2</sub>O-N g<sup>-1</sup> f.w. h<sup>-1</sup>, is at the lower end of this range.

Although the direct N<sub>2</sub>O emissions have been quantified in many studies, there are few estimations of their proportion of total emissions. In our laboratory experiment, the proportion emitted by *L. terrestris* of the total N<sub>2</sub>O flux was on average 16 %, which is in good agreement with that reported by Karsten and Drake (1997) for beech forest soil (16 %), but significantly higher than their value for oak-beech forest soil (0.25 %). Our estimate is high and it may overestimate the proportion in the field because the time interval *L. terrestris* was able to shape the soil was short in our laboratory trial. In the field, the soil is subjected to a long-term earthworm impact and it is likely that this leads to a greater contribution of indirect emissions from the environment. It should also be noted that part of the N<sub>2</sub>O produced by the earthworms may be reduced to N<sub>2</sub> while diffusing from the soil to the atmosphere and the significance of direct emissions may also for this reason in the field be lower than estimated based on laboratory measurements. Consequently, it is likely that the enhanced N<sub>2</sub>O emissions in the presence of *L. terrestris* are also due to the changes in topsoil conditions and creation of hot spots of high biological activity, including the elevated macrofaunal densities, in the vicinity of the middens. For instance, the higher content of mineral nitrogen and soil moisture favour denitrification, which was manifested as elevated values of potential denitrification in our measurements. In our field site, soil moisture was nearly 40 %, corresponding to 80 % WFPS, which is suitable for earthworm N<sub>2</sub>O contribution (Evers et al., 2010). Another potential mechanism for increased N<sub>2</sub>O emissions in the field are the burrows that may act as large pores that ease the diffusion of N<sub>2</sub>O from the bottom soil and allow more of the N<sub>2</sub>O ending up in the atmosphere without being reduced to N<sub>2</sub>. The laboratory soil was dryer than the field soil, which could be one reason for the less noteworthy earthworm effect as soil moisture can significantly modify the earthworm-induced N<sub>2</sub>O emissions (Chen et al., 2014).



**Table 5.** Characteristics of *L. terrestris* ( $n = 13$ ) and control mesocosms ( $n = 15$ ) at the end of the laboratory experiment (model-based mean estimates and 95 % confidence intervals presented for all variables). *F*- and *P*-statistics show the statistical significance of the difference between the *L. terrestris* and control mesocosms.

	<i>L. terrestris</i>	Control	<i>df</i>	<i>F</i>	<i>P</i>
Mineral N (mg kg <sup>-1</sup> soil d.w.) <sup>a</sup>	21.9 (20.9–23.0)	21.0 (20.0–21.9)	1, 12.3	8.71	0.012
Soil moisture (% of f.w.) <sup>b</sup>	20.7 (20.6–20.8)	20.4 (20.3–20.5)	1, 14.1	13.46	0.003
Mineral N (mg kg <sup>-1</sup> soil d.w.) <sup>b</sup>	23.1 (21.0–25.4)	19.3 (17.6–21.2)	1, 24	7.74	0.010
Potential denitrification (µg N <sub>2</sub> O–N g <sup>-1</sup> soil d.w. h <sup>-1</sup> ) <sup>b</sup>	0.30 (0.27–0.32)	0.25 (0.23–0.27)	1, 26	10.55	0.003

<sup>a</sup> Sample represents the entire soil column (excluding the soil core). <sup>b</sup> Soil core (depth 5 cm, diameter 5 cm) in the *L. terrestris* mesocosm taken around the burrow entrance.

The increase in soil cumulative CO<sub>2</sub> emissions due to the presence of *L. terrestris* was 33 and 15 % in our field and laboratory measurements, respectively. These results echo the meta-analysis by Lubbers et al. (2013a), which suggests a 33 % increase in soil CO<sub>2</sub> emissions in the presence of earthworms. When we estimated the respiration of individual earthworms in the laboratory, the mean CO<sub>2</sub> emission (425 µg h<sup>-1</sup>) was almost double the mean difference between the mesocosms with and without *L. terrestris* (230 µg chamber area h<sup>-1</sup>) and the proportion of the total CO<sub>2</sub> flux explained by earthworm respiration was 36 %. These values suggest that the increased emissions of CO<sub>2</sub> from the soils occupied by *L. terrestris* were fully explainable by the respiration of the animal itself. If this is true in general, the discrepancy between the observations of increased CO<sub>2</sub> emissions vs. increased carbon stability (Lubbers et al., 2013a) would be explained by earthworm respiration counteracting the enhanced carbon sequestration. However, this conclusion has to be treated cautiously as we do not know how well the measurements of earthworm respiration in the laboratory represent the respiration in the field. In the field, the elevated slug densities of the middens also likely contributed to increased CO<sub>2</sub> emissions as snail castings and mucus have been observed to increase the efflux from leaf litter (Theenhaus and Scheu, 1996). Snail activity accelerates N cycling, too (Theenhaus and Scheu, 1996), but we are not aware of any studies of snail impacts on N<sub>2</sub>O emissions.

Unlike the effects on N<sub>2</sub>O and CO<sub>2</sub> fluxes, the effects of *L. terrestris* on CH<sub>4</sub> flux were variable and mostly inconsequential and there was only a slight indication in the cumulative field fluxes that the presence of *L. terrestris* might decrease soil CH<sub>4</sub> oxidation rate. Such a decrease could be a consequence of increased moisture and N content in the vicinity of middens (Hütsch, 2001). Small and varying earthworm effects on net CH<sub>4</sub> fluxes have also been reported earlier (Borken et al., 2000; Aira et al., 2009; Bradley et al., 2012), and our estimate of 0.7 % *L. terrestris* contribution to the total CH<sub>4</sub> flux is in good agreement with the earlier statement that *L. terrestris* is not a source of CH<sub>4</sub> (Šustr and Šimek 2009). As CH<sub>4</sub> fluxes are also in general non-significant in the context of carbon cycling in boreal arable soils (Regina

et al., 2007), it appears that the effects of earthworms on the GWP of these soils are driven by their effects on N<sub>2</sub>O and CO<sub>2</sub> emissions.

Recent studies suggest that Finnish no-till fields are characterized by both high population densities of *L. terrestris* (Nuutinen et al., 2011) and elevated N<sub>2</sub>O emissions (Sheehy et al., 2013). Higher N<sub>2</sub>O emissions are usually explained by denser soil structure and higher soil moisture compared to tilled soils. Our results suggest that increased population densities of *L. terrestris* can also contribute to the elevated N<sub>2</sub>O emissions. We found on average 20 *L. terrestris* middens per m<sup>2</sup> in our no-till field and when compared to a square metre of equal field with no middens, such a density would increase the N<sub>2</sub>O emissions by 27 % (estimated using mean values of midden and non-midden areas). Although this estimate has to be treated with caution as the non-midden areas were not completely out of the reach of *L. terrestris* activity, it appears that enhanced earthworm activity may explain a substantial part of the 60–150 % increase in N<sub>2</sub>O emissions observed in Finnish no-till fields (Sheehy et al., 2013). Moreover, when all three gases were considered together, *L. terrestris* increased the GWP of the soil by 50 and 18 % in our field and laboratory investigations, respectively. These values, and particularly the field estimate, exceed the 16 % mean increase in the net GWP of laboratory soils reported by Lubbers et al. (2013a) in their meta-analysis based on 33 observations from individual earthworm studies that reported the cumulative emissions of both N<sub>2</sub>O and CO<sub>2</sub>. However, the temporal variation in emissions is probably high, mainly due to soil moisture variation. For example, in a field study by Lubbers et al. (2013b), earthworms increased N<sub>2</sub>O emissions of managed grassland in the autumn when the WFPS of soil was 61–65 %, but had no effect in the dry spring when the WFPS was 16–25 %. Our field experiment represents the conditions that prevail for approximately 3 months in the autumn when *L. terrestris* is highly active and it is possible that during other seasons, the gas emissions are less affected by the species. Moreover, the field estimate may exaggerate the earthworm effect as part of the straw in the non-midden areas and was likely transferred and consumed in the midden area. In contrast to what we expected, the contributions of

earthworm-induced N<sub>2</sub>O and CO<sub>2</sub> emissions to the net increase in GWP were 6 and 94 % in the field and 2 and 98 % in the laboratory, respectively. This indicates that the elevated N<sub>2</sub>O actually has a minor significance in the total balance despite its high GWP value.

One of our aims was to test whether the earthworm effects on GHG emissions that are found in laboratory trials can be generalized to field conditions. For this purpose, we established a mesocosm experiment using soil and *L. terrestris* individuals collected from the field site. The mesocosms had generally higher CO<sub>2</sub> and lower N<sub>2</sub>O emission rates than the field soil, which probably was due to soil sieving increasing the availability of microbial resources and microbial respiration (Hartley et al., 2007) and drier mesocosm soil supporting lower N<sub>2</sub>O production. Unlike in the field, the flux rates also steadily decreased in the laboratory, which probably indicates diminishing resource availability after the initial resource pulse (Hartley et al., 2007). Despite these differences in the level and dynamics of the flux rates, a clear, positive effect of *L. terrestris* on N<sub>2</sub>O and CO<sub>2</sub> emissions was found in both systems. The magnitude of the *L. terrestris* effect was smaller in the laboratory, which could be related to soil moisture and the loss of earthworm weight over the experiment, but also to the significantly elevated faunal abundance and activity in the long-lived *L. terrestris* living sites in the field. The size of the effect on CO<sub>2</sub> emissions also decreased in the laboratory as the experiment proceeded. Such a decrease is common in laboratory studies (Borken et al., 2000; Lubbers et al., 2013a) and is most probably related to the lack of fresh plant input to the soil, which has a negative impact on *L. terrestris* metabolism. The distinct difference between the field and laboratory emissions in their response to the removal of middens and residues from the soil surface can possibly be explained by the lack of air current in laboratory conditions, which may have led to GHG accumulation in the soil pores and release of gases when the midden and straw were removed. All these findings suggest that while the general influence of *L. terrestris* on GHG emissions can be approximated in laboratory conditions, field measurements are needed for more accurate estimates and proper mechanistic understanding.

To conclude, our study contributes to filling the gap of field studies of the effects of earthworms on GHG emissions, particularly in soils long occupied by earthworms (Lubbers et al., 2013a). Our results emphasize the significance of *L. terrestris* in the gas balance of agricultural soils, and especially in no-till fields. We showed that *L. terrestris* respiration can explain the observed increase in CO<sub>2</sub> emissions in the presence of earthworms and that a substantial part of the increase of N<sub>2</sub>O emissions in no-till arable lands can be explained by earthworm contribution. The gap of knowledge that still remains after our study is that the effects of earthworms have almost solely been studied in the absence of plants and without considering plant growth. As the effects of earthworms on plant growth are generally positive (van Groenigen et al.,

2014), the disservice of increased N<sub>2</sub>O emissions may be counteracted by enhanced plant growth to the degree that no increase in yield-scaled emissions results (Wu et al., 2015). Extrapolation from our results to field scale may not be simple either as the effect of midden density on GHG production is not necessarily linear due to resource competition among earthworm individuals. However, considering that field soils with active *L. terrestris* middens had 50 % higher global warming potential than non-midden areas, it is clear that *L. terrestris*, and potentially other earthworm species as well, are among the key players that need to be taken into consideration when the role of agricultural soils and cultivation practises are evaluated for climate change mitigation. All in all, our study points out how studies on the effects of conservation practices are necessary to fully understand their effects on the environment.

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